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Case Report

PFAS contamination in soil and sediment: Contribution of sources and environmental impacts on soil biota

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ABSTRACT

Per- and polyfluoroalkyl substances (PFAS) have been detected in soils worldwide, suggesting that soil serves as a substantial reservoir for these compounds. This paper provides a comprehensive overview of the origin of PFAS contamination in soil and sediment, as well as the environmental impacts associated with the distribution of PFAS in these mediums. In this review, we also discuss the effects of PFAS on soil microbes, plants, the terrestrial food web, sediment microbes, and benthic microorganisms. The highest concentration of PFAS was found in soils exposed to aqueous film-forming foam (AFFF), the PFAS manufacturing plants, and biosolids, compared to landfills, atmospheric deposition, and industrial discharge, which are the other three sources. Exposure to PFAS has demonstrated negative effects on the richness, diversity, and population structure of bacterial species. It has been observed to interfere with the regular functioning of soil microorganisms, microbes, and benthic organisms in sediment. Nevertheless, the current data is insufficient to grasp the complete spectrum of effects comprehensively. Moreover, the accumulation of PFAS in plants, vegetables, and earthworms is a cause for concern, as they can eventually enter the food chain through trophic transfer processes. Finally, we discuss the challenges and possible strategies that can be implemented to restrict the dispersion of PFAS in sediments and soil.

1. Introduction

PFAS are widely distributed throughout the environment, with detectable levels in most soils [1–3] and streams. These compounds are present in numerous forms, depending on nearby sources [4]. Consequently, PFAS have been found to be associated with various adverse effects in both aquatic species and humans [5]. PFAS have been detected in regions with no apparent signs of human-related activities that could serve as a potential source [6]. A comprehensive assessment of the sources and distribution of PFAS in a specific location is essential to make well-informed decisions about remediation. The occurrence, movement, and ultimate destiny of PFAS in natural surroundings have been extensively documented on different scales, incorporating specific contaminated areas, regional investigations, and global assessments [7]. Through a variety of pathways, PFAS can enter the soil ecosystem. PFAS have already been frequently identified in soils in various quantities [3]. Although direct exposure from the consumption of manufactured goods and drinking water can be effectively controlled, preventing indirect exposure through accumulation in freshwater and terrestrial sources of food is more challenging [8]. Furthermore, this characterization is crucial to understanding the mechanisms involved in the fate and movement of PFAS [9].

There has been a widespread occurrence of PFAS contamination in various abiotic and biotic systems around the world [10]. Based on life cycle evaluations conducted on commercial PFAS manufacture between 1951 and 2002, researchers have approximated that 2600–5000 tonnes

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of C_4–C_{14} PFAS have been discharged. Furthermore, it is projected that from 2015 to 2030, a maximum of 5510 tonnes of C_4–C_{14} PFAS might be released into the environment [11]. Consequently, studies conducted in the vicinity of PFAS contaminated areas have revealed that various food items, such as crops [12], cereals, vegetables [13,14], fruits [15], as well as milk and meat [16], sourced from these locations may contain elevated levels of PFAS compounds. In addition, the presence of PFAS can have implications for interactions among microorganisms within soil ecosystems [7]. Uptake of PFAS by plants from the soil and subsequent transfer to food chains can pose a concern to human health [17]. The proliferation of large-scale manufacturing and consumption has resulted in the widespread presence of PFAS in aquatic habitats and its notable accumulation in the sediments of benthic ecosystems [18]. These compounds have the ability to negatively impact the well-being of microbial species and communities within ecosystems [19,20]. The anticipated reduction in microbial diversity and activities caused by harmful substances is expected to influence essential ecological processes, transmitting adverse impacts to higher trophic levels. Consequently, this will have implications for sediment environments' long-term viability and ability to recover from disturbances [21]. Therefore, research is imperative on the accumulation of PFAS in soil, sediment, and other organisms and their environmental repercussions.

To address this need, our study aims to provide a comprehensive investigation of the origins of PFAS in soil and sediment, comparing contributions from various sources (Table 1). Furthermore, the present study examines the spatial distribution based on the types of sources, the leaching and transport mechanism of PFAS from soil, and the environmental consequences of PFAS when introduced into soil and sediment matrices. Two recent academic articles have been published, one by Li and Koosaletse-Mswela, (2023) [22], discussing the sources of PFAS in soils, their sorption and transport characteristics, and various remediation options. The other publication by Brusseau et al. (2020) focuses on PFAS concentrations in soils. At the time of writing this article, we understand that there is a lack of comprehensive reviews of the literature on the environmental consequences of PFAS in soil and sediment.

2. Contribution of sources in contamination of soil and sediment

PFAS pollution is predominantly caused by human activity and originates from local sources such as PFAS manufacturing plants, discharges [47], landfills, air emissions, airports, military bases, consumer consumption, and fire training areas (Fig. 1) [48]. For example, both short- and long-chain perfluoroalkyl acids (PFAAs) were extensively found in the influents of wastewater treatment plants (WWTPs) at concentrations of up to 1000 ng/L. Similarly, these compounds were also detected in WWTP effluents, with concentrations ranging from 15 to more than 1,500 ng/L [49]. There are three methods for dealing with PFAS waste: incineration, wastewater treatment, and landfilling. However, it is important to note that conventional wastewater treatment and landfilling merely relocate PFAS burdens from one site to another without effectively destroying them [50]. Lenka et al. (2021) [49] found that most traditional wastewater treatment plants (WWTPs) had poor performance in removing PFAS. Furthermore, several studies indicated an elevation in the levels of polyfluoroalkyl acids (PFAAs) after wastewater treatment. This increase was attributed to the conversion of PFAS precursors to PFAAs within WWTPs. Advanced processes such as adsorption using ion-exchange resins, electrochemical degradation, and nanofiltration are highly effective in removing PFAS, with approximately 95–100 % removal rates. However, the implementation of these treatments in real-world WWTPs faces challenges, particularly when it comes to scaling up. Several entry pathways, including direct release and breakdown of precursors, contributed to PFAS in soils and sediment at various locations [4]. For example, compounds of perfluorinated ether carboxylic acid and perfluorinated ether sulfonic acid, featuring one or two acidic functional groups, have been detected in sediment at various sites in the upper region of the Cape Fear River in North Carolina, USA. The source of this contamination is believed to be a Chemours PFAS manufacturing facility located close to the Cape Fear River [51]. In addition to direct soil pollution from primary sources, there is growing concern about the contribution of indirect sources, including transformations by precursor components or atmospheric deposition

Table 1
Concentration and potential sources of PFAS in soil and sediment (ng/g).

Location	Media	Potential sources	PFAS	Concentration range	Reference
Shanghai, China	Soil	AFFF	Σ_{11}PFAS	3.4–531.7	[23]
Norway	Soil	AFFF	Σ_{12}PFAS	0.3–6500	[24]
Norwegian airport	Soil	AFFF	PFOS	9–2600	[25]
Australia	Soil	AFFF	PFOS	0.2–1400	[26]
Finland	Soil	AFFF	Σ_{23}PFAS	2800	[27]
Xiaoqing River, China	Sediment	Fluoropolymer production facility	Σ_{12}PFAA	8.44–465.6	[28]
Fujian, China	Soil	POSF manufacturing facilities	Σ_{18}PFAS	26300	[29]
Shandong, China	Soil	Fluoropolymer production facility	Σ_{18}PFAS	4.40–189	[30]
Beijing, China	Soil	Biosolids	Σ_{11}PFAS	41.4–220	[31]
USA	Soil	Biosolid based compost	Σ_{17}PFAS	9.0–199	[32]
Australia	Soil	Biosolids	Σ_{38}PFAS	10.4–104	[33]
USA	Soil	Compost	Σ_{17}PFAS	28.7–75.9	[34]
Shanghai, China	Soil	Industrial areas, airports, landfills, fire stations and agricultural areas	Σ_{18}PFAS	0.64–294	[4]
Sichuan, China	Soil	Industrial city	Σ_{11}PFAS	0.18–1.85	[35]
South Korea	Soil	Industrial complexes	Σ_{15}PFAS	0.346–11.7	[36]
Jucar River, Spain	Sediment	Industrial and urban discharges	Σ_{9}PFAS	0.22–11.5	[37]
Michigan, USA	Sediment	Airports and urban areas with heavy industry	Σ_{10}PFAA	0–3.25	[38]
Pensacola, USA	Sediment	Military bases, airports, carpet factories	Σ_{51}PFAS	0.04–0.48	[19]
Minnesota, USA	Soil	Irrigation water	Σ_{7}PFAS	0.71–29.64	[39]
Fuxin, China	Soil	Irrigation water	Σ_{3}PFAS	58.9	[10]
North America	Soil	Landfill	Σ_{32}PFAS	0.18–8.07	[2]
Ireland	Soil	Landfill	Σ_{5}PFAS	0.15–6	[40]
China	Soil	Landfill	Σ_{17}PFAS	5.31–108	[41]
South Korea	Soil	Landfill	Σ_{15}PFAS	0.5–10.4	[36]
Ohio, USA	Soil	Waste Incineration	Σ_{8}PFAS	0.05–8.3	[42]
North Carolina, USA	Soil	Atmospheric deposition near fluoropolymer production plant	PFOA+ HFPO−DA	30.1	[43]
Netherlands	Soil	Atmospheric deposition near fluoropolymer production plant	PFOS + PFOA	3.3	[44]
France	Soil	Atmospheric deposition near fluoropolymer plant	Σ_{11}PFAS	42–347	[45]
Bering Sea	Sediment	Atmospheric deposition, ocean waves, regional supplies	Σ_{9}PFAS	0.06–1.73	[46]

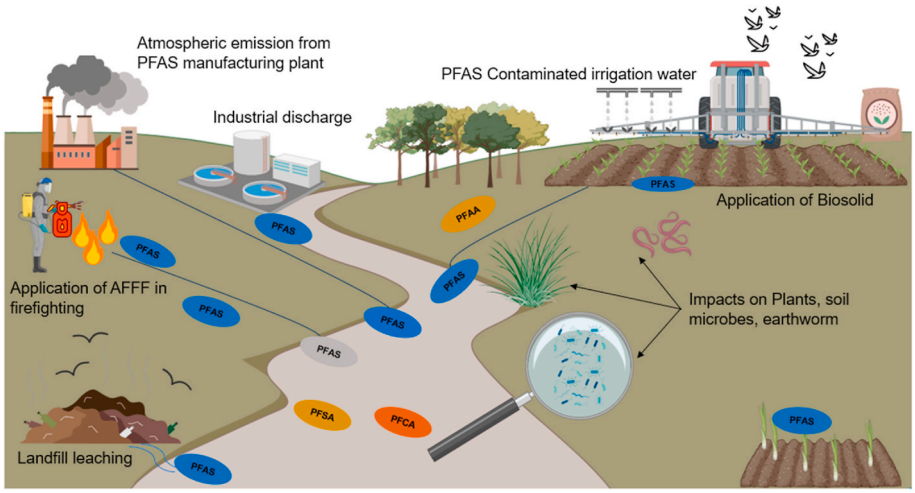

Fig. 1. Potential sources, pathways, and impacts of PFAS in soil and sediments. The figure illustrates the main sources that are responsible for the contamination of soil and sediments with PFAS. Atmospheric emissions and discharges from PFAS manufacturing plants, application of biosolids, and application of AFFF in fire-fighting are the principal sources of contamination in soil.

[36]. PFAS can also accumulate in regions far from human activity through atmospheric deposition due to long-range transport [36]. Multiple studies have documented the presence of PFAS in wet precipitation, highlighting that wet deposition is the predominant mechanism for the transfer of PFAS from the atmosphere to terrestrial environments [52]. For example, sediment samples from Canada's remote northern regions, such as Lake Hazen and Lake B35, show that these areas are affected by long-range transport and several emission sources for PFAS [53]. Furthermore, sediment desorption from tributaries may have an impact on PFAAs transport [38]. Therefore, sediment has the potential to serve as a source of PFAS when water concentrations are reduced through pollution control techniques.

2.1. Frequently examined PFAS types

PFAS are found in many families, each of which has a large number of homologous members and isomers. In general, PFAS are classified as either long chain ($C \geq 8$) or short chain ($C < 8$), depending on the number of fluorinated carbon atoms in their backbone, complete or partially fluorinated alkyl chain [54]. Perfluoroalkyl acids (PFAAs) are widely recognized in the family of PFAS literature, and the PFAAs include perfluoroalkyl carboxylic, sulfonic, sulfinic, phosphonic, and phosphinic acids. Perfluoroalkyl carboxylic acids $C_nF_{2n+1}COOH$ (PFCA) and perfluoroalkyl sulfonic acids $C_nF_{2n+1}SO_3H$ (PFSA) are two prominent families of PFAA that are extensively studied in scientific research. Moreover, the most frequently mentioned perfluorinated carboxylic acid (PFCA) is perfluorooctanoic acid (PFOA); however, perfluorooctane sulfonic acid (PFOS) is frequently discussed under PFSA [55]. In addition to PFOA and PFOS, the most commonly researched PFAAs based on their carbon content are Perfluorobutanoic acid (PFBA), Perfluoropentaonic acid (PFPeA), Perfluorohexaonic acid (PFHxA), Perfluoroheptaonic acid (PFHpA), Perfluorononaonic acid (PFNA), Perfluorodecaonic acid (PFDA), Perfluoroundecanoic acid (PFUnA), Perfluorododecanoic acid (PFDoA), Perfluorotridecanoic acid (PFTrDA), and Perfluorotetradecanoic acid (PFTeDA). Similarly, perfluorobutane sulfonic acid (PFBS), Perfluororpentane sulfonic acid (PFPeS), perfluorohexane sulfonic acid (PFHxS), Perfluoroheptane sulfonic acid (PFHpS), Perfluorononane sulfonic acid (PFNS), and perfluorodecane sulfonic acid (PFDS) are commonly documented perfluoro sulfonic acids (PFSA). Besides perfluoroalkyl compounds, perfluoroalkane sulfonamido substances (eg N-Alkyl perfluoroalkane sulfonamides, EtFASA), Perfluoroalkane sulfonamides (e.g. Perfluorooctane sulfonamide, FOSA) and fluorotelomer substances (eg n:2 Fluorotelomer alcohols, FTOH) sometimes get documented.

2.2. Biosolids

Agricultural biosolids, considered a valuable resource, are under growing scrutiny as potential sources of PFAS in soil and water bodies [11]. PFAS can be found in municipal biosolids and treated wastewater sludge even after extensive treatment [6]. Disposing wastewater sludge through landfill or incineration and using biosolids in agricultural fields pose a possible risk of contamination of soil, water, and agricultural fields with PFAS [56]. Incineration of PFAS containing materials can release hazardous air pollutants. The United States annually generates 7.2 million dry metric tons of biosolids, and approximately 55 % of this amount, equivalent to 3.9 million tons, is applied to the soil [57]. Moreover, the average quantity of ∑PFAS found in U.S. biosolids was calculated to be between 2749 and 3450 kg per year. Of this amount, about 1375–2070 kg were used for agricultural purposes, while 467–587 kg were sent to landfills as an alternative method of disposal [58]. An investigation of the PFAS content of agricultural land in southern Arizona, after more than 30 years of soil fertilization with municipal biosolids, revealed that these modified biosolid soils have mean PFAS contents ranging from undetectable to 4.1 ng/g. However, soil samples obtained from undisturbed areas with no record of agricultural activity, irrigation, or biosolids did not show the presence of PFAS compounds [59]. Similarly, an agricultural facility with a long history of biosolid treatments detected 12 PFAS in surface soils, in the vadose zone, and groundwater [11]. It is important to note that biosolids derived from industrial sources have the potential to contribute significantly to PFAS contamination, particularly compared to municipal biosolids [6]. For example, the concentrations of PFAS in agricultural byproducts, including pig slurry, poultry manure, and dairy cattle manure, were found to be relatively low (median ∑46 PFAS: 0.66 ng/g dry matter). Conversely, urban and industrial wastes, paper mill sludge, sewage sludge, and residual household waste composts contained higher levels of PFAS (median ∑46 PFAS: 220 ng/g) [60].

2.3. Aqueous film-forming foam (AFFF)

Given the widespread use of AFFF and the high levels of contamination detected at numerous fire training sites in the United States, Finland, and other countries, it was evident that the fire station was to be treated as a point source for direct emission of PFAS [4]. Therefore, this emission has resulted in the introduction of PFAS into soil, groundwater,

and surface water [9,61]. The infiltration of PFAS into shallow soils with limited permeability is believed to have occurred primarily from the surface. As a result, the gradual back-diffusion process can affect the subsequent release of PFAS into groundwater [9]. For example, high levels of PFAS were discovered in soil in a Norwegian firefighting training center 15 years after PFOS-based AFFF was introduced [24]. Additionally, 96 % of the overall PFAS concentrations in the soil were found to be PFOS, with values ranging from 0.3 ng/g to 6500 ng/g in that area.

2.4. Compost, animal, and poultry manure

Compost is a by-product derived from the waste recycling process known as composting that adds value to the broader agricultural process. Food waste, lawn debris, and food packaging components are all parts of municipal solid waste that both the commercial and residential sectors produce and use for composting. Soils with inadequate texture and nutrient content can be improved by incorporating various organic amendments, including readily available composts, garden soils, and natural potting mixes. These products are widely available in consumer markets around the world and serve to facilitate optimal plant development. Elevated concentrations of PFAS were observed in municipal and industrial waste materials, paper sludge, sewage effluent, and leftovers from household composts. Therefore, the median total PFAS concentration (46 PFAS) was measured to be 220 ng/g in these composts [62]. In another study, the mean concentration of 38 PFAS in compost in Australia was found to be 8.89 ng/g [63]. In the United States, various types of non-biosolid organic composts, including food, yard, manure, and mushroom-based materials, available on the market, exhibited total concentrations of 17 PFAS ranging from 28.7 to 75.9 ng/g [34]. Both PFOA and PFOS were found in all compost samples. However, most composts were mainly composed of short-chain PFAA, representing more than 64 % of the total composition. In a separate investigation, the measured concentration of 17 PFAS in biosolid-based products ranges from 9.0 to 199 ng/g, which is higher compared to food and yard waste at 18.5 ng/g and other organic products at 0.1–1.1 ng/g [32]. PFAS were found to be present in relatively low concentrations in agricultural waste by-products, including pig slurry, chicken manure, and dairy cattle dung. The median total concentration of 46 PFAS compounds in these wastes was measured at 0.66 ng/g of dry matter. In summary, a higher concentration of PFAS was observed in the compost materials, and the level of contamination by PFAS depended on the specific types of raw materials used in the compost production process.

2.5. Landfill leachate

Households, businesses, and manufacturing industries produce substantial solid waste [64], including construction and manufacturing industries, building waste, biosolids from wastewater treatment plants (WWTPs), industrial waste, and debris resulting from disasters. Products containing PFAS are disposed of at the end of use [65], contributing to the significant handling of large amounts of PFAS through the solid waste system [66]. Estimates indicate that the total leachate produced in the United States was 61.1 million m^3, with approximately 79 % of this volume originating from landfills located in regions with wet climates [65]. Estimation of the discharge of PFAS to landfills is a complex task due to the wide range of quantifiable PFAS compounds [64]. Nevertheless, Rankin et al. (2016) [2] investigated the spatial distribution of 32 PFAS in surface soils from 62 different locations, encompassing all continents. All samples were found to contain measurable amounts of PFCA, specifically PFHxA and PFTeDA. The total quantities of PFCA ranged from 0.029 to 14.3 ng/g (dry weight). Furthermore, the diversity of landfill waste further complicates this estimation process. There are multiple routes for releasing PFAS from landfills into the soil, most commonly in the form of leachate from contaminated waste. In addition to ventilation, gas flares are another potential source of atmospheric PFAS exposure, which ends up in soil and surface water.

2.6. PFAS manufacturing plants and industrial discharge

PFAS manufacturing facilities had the highest levels of PFAS detection in nearby locations and other places (Table 1). Additionally, soil samples collected from industrial sites showed the highest concentrations of PFAS in areas related to textile manufacturing, followed by metal, chemical and electronic manufacturing regions [36]. Soil, surface water, and groundwater contamination from air deposition near or far from PFAS manufacturing facilities could endanger people and the environment [43]. Shimizu et al. (2021) [67] found that six PFAS compounds had an overall concentration that varied within the quantification limit to 110 ng/L by wet deposition and overall discharges of 0.3–29 ng/m^2/day through dry deposition near a PFAS manufacturing facility in North Carolina. All six PFAS were expected to release 30 μg/m^2 of wet deposition and 1.4 μg/m^2 of dry deposition each year. In Wisconsin, the daily flux values of PFAS ranged from 1.3 to 47.4 ng/m^2/day, with a median value of 5.7 ng/m^2/day. The total levels of PFAS mass in rainfall varied between 0.7 and 6.1 ng/L, with a median of 1.5 ng/L. Evaluation of air flux is crucial to establish and validate measured PFAS in the cores of sediment and soil [68]. In another study, 13 soil samples were collected near a fluoropolymer production plant located in Parkersburg, West Virginia. The samples, collected in the upstream and downstream areas of the facility, exhibited diverse amounts of perfluorooctanoic acid (PFOA) ranging from 5 to 27 ng/g during the period 2016–2018 [43].

Numerous studies have documented the amount of PFAS in soil and sediments while considering different PFAS sources. In this review, we identified six primary sources of PFAS that persistently polluted soil and sediment. Not all sources have the same level of contamination or severity. The data in Table 1 reveal that aqueous film-forming foam (AFFF) originating from the fire training area, the fluoropolymer production plant, and biosolids are the three main contributors of PFAS in soil.

3. Distribution of PFAS in soil and sediment based on source types

Soil and sediment serve as reservoirs for various anthropogenic contaminants, such as PFAS [69,70]. The degree of pollution in a given region depends on the specific sources of PFAS in that location. For example, Cao et al. (2022) [23] identified the presence of PFAS compounds in soil samples collected from seven plots at fire training locations. They found that PFAS concentrations ranged from 3.4 to 531.7 ng/g dry weight. These sites had been using PFOS-restricted aqueous film-forming foams (AFFFs) for a period of 6 years. On the other hand, the Minnesota Department of Health investigated the presence of PFAS in outdoor water supplies, garden soil, and garden produce. The objective was to examine the occurrence of elevated concentrations of PFAS resulting from the use of contaminated garden irrigation water. PFBA, PFOA, and PFOS were observed in all soil samples, exhibiting higher concentration levels compared to other PFAS [39]. In another investigation, soils that received biosolid five times between 2007 and 2020, at a rate of 6–9 dry tons/ha of biosolid cake, exhibited a total concentration of 38 PFAS that ranged from 10.4 to 104 ng/g (dry weight) [33]. The PFOS concentration in soils was found to be the highest, reaching a maximum value of 48.1 ng/g, and N-ethylperfluoro-1- octanesulfonamidoacetic acid (N–Et-FOSAA) closely follows at a maximum concentration of 10.9 ng/g. In another study, soils supplemented with biosolids were identified with a total of nine perfluorocarboxylic acids (PFCA) and three perfluorosulfonic acids (PFSA). PFCA concentrations ranged from 18.0 to 113 ng/g, while PFSA concentrations ranged from 23.4 to 107 ng/g [31].

Martin et al. (2023) [42] examined the distribution of PFAS in soil samples around a waste incinerator. Analysis of 35 soil samples revealed

measurable amounts of PFAS, including PFBS, PFOS, PFOA, and GenX, with observed concentrations ranging from 0.05 to 8.3 ng/g. They also concluded that incineration effectively eradicated a significant proportion of PFAS, while incomplete incineration led to the generation of by-products. In another study, surface sediment samples were taken from the Xiaoqing River near Shandong Province of China, near a prominent fluoropolymer company. The analysis of these samples aimed to assess the potential contamination resulting from the company's activities. The total concentration of PFAA in the sediment layers varied between 8.44 and 465.6 ng/g (dw), with an average concentration of 109.47 ng/g (dw). PFOA emerged as the predominant PFAA, exhibiting a range of concentrations of 3.86–456.20 ng/g (dw), with an average concentration of 103.14 ng/g of dw [28]. Similarly, the occurrence of 51 PFAS was assessed by testing sediment samples from 25 sites in the Pensacola Bay System, Florida watershed. PFBA was detected in all sediment samples, with concentrations ranging from 0.04 to 0.48 ng/g (dw); the location with the highest level of total PFAS was 3.89 ng/g. Military installations, airports, carpet factories, paper mills, farms, and forests in that study area were potential sources of PFAS [19]. Balgooyen and Remucal (2022) [38] evaluated the levels of ten PFAA in sediment samples collected from 41 tributaries in the Green Bay region of Michigan. PFAA concentrations ranged from below the detection limit to 3.25 ng/g, averaging 0.73 ng/g. Airports and urban areas with intensive industrial activities were identified as potential sources of this contamination. Similarly, the Jucar River in Spain was found to have average concentrations of PFAS in its sediment ranging from 0.22 to 11.5 ng/g as a result of industrial and urban discharges [37]. The sediment of the Bering Sea and the northeast Chukchi Sea in the Arctic region exhibited predominant PFOS, PFNA, and PFBS levels, with total PFAS values ranging from 0.06 to 1.73 ng/g (dw). The primary origins of PFAS in Arctic sediment can be linked to the degradation of perfluorooctane sulfonyl fluoride (POSF) in consumer goods, the production of fluoropolymers, and the release of AFFF. These sources are mostly introduced into the Arctic environment by atmospheric deposition, ocean waves, and regional supplies [46].

Here, we observed the concentration of PFAS in soil and sediment originating from different sources, such as a PFAS manufacturing plant, the use of AFFF in a fire training area, the use of biosolids in soil, contaminated irrigation water, air deposition, waste incineration and discharges from factories and cities (Fig. 1). The release of PFAS from these various sources has resulted in the contamination of soil, sediment, surface water, and groundwater. This contamination occurs through processes such as air emissions, discharges into water bodies, leaching, incineration, and runoff from wastewater (Fig. 2). Table 1 reveals a significant variation in the concentration of sources of PFAS, depending on the types of PFAS and also on geographic location. China, in particular, exhibits a higher degree of PFAS contamination in its soil. The area affected by the application of AFFF, PFAS manufacturing plants, and biosolids had the highest PFAS concentration in the soil compared to three other sources: landfills, atmospheric deposition, and industrial discharge. Therefore, it is important to carefully monitor these three sources because they could potentially contaminate the soil with PFAS. Our findings are consistent with the study conducted by Zhu et al. (2022) [4] in China, which reported that PFAS concentrations and detection frequencies were highest in fire stations, followed by industrial sites such as waste disposal plants, electroplating facilities, and the chemical sector. In addition, PFAS concentrations and detection frequencies were also shown to be elevated in other regions, such as landfills, airports, and farmlands.

4. Leaching and transport of PFAS in soil

Leaching of PFAS in the soil is a crucial mechanism that contributes to groundwater contamination, making it imperative to understand its ultimate destiny and ecological consequences [71]. PFAS, being highly persistent and mobile, frequently migrates from the soils to surrounding water bodies, including groundwater [72]. The main factors that influence the fate of introduced PFAS into the soil environment are the infiltration and sorption properties on the solid substrate. In polluted agricultural soils, Röhler et al. (2021) [73] investigated the long-term fate of PFAS. In particular, they examined seasonal variation in leaching patterns in agricultural soils, predicted extended leaching of PFAS over a long period of time from polluted agricultural areas, and finally quantified the persistence of PFAS from agricultural sources. 12-years of PFAS site data showed that short-chain PFAS were produced and accumulated during summer and spring and mobilized in the fall and winter. The results of soil leaching studies indicate that PFAS entering the groundwater from the topsoil can span several months or even years [29]. The complete removal of PFAS from the surface soil is expected to last for several decades. Furthermore, all sites exhibited elevated quantities of long-chain PFAS, predominantly distributed in the topsoil. For example, long-chain PFAS, namely, perfluoroundecanoic acid (PFUdA) and perfluorotetradecanoic acid (PFTeDA), exhibited a preference to remain in the uppermost layer of the soil. In contrast, short-chain PFAS, including perfluorobutanoic acid (PFBA), perfluoropentanoic acid (PFPeA), and perfluorohexanoic acid (PFHxA), showed a tendency to migrate to a depth of about 100 cm within the vertical profile of the soil [35]. PFAS showed a downward migration at various soil depths over time, with a general pattern of decreasing concentrations from the top of the soil core to its base [29]. Much lower PFAS loading were likely to be found in runoff on soils with good drainage since PFAS mass transfer occurred mostly downwards through the soil structure by leaching [74]. However, in soils with inadequate drainage, which can be caused by compaction, efficient leaching may not be possible and PFAS are transported mainly by runoff [74]. Hydraulic parameters, such as infiltration and evapotranspiration rates, along with soil characteristics and the particular characteristics of PFAS compounds, collectively influence the transport or migration of PFAS

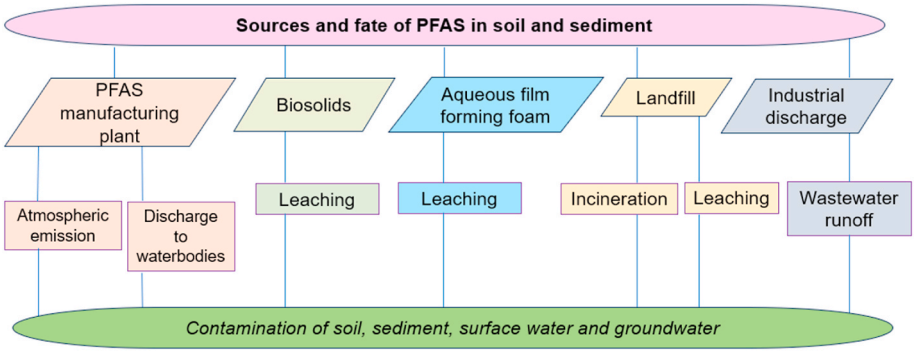

Fig. 2. Sources and fate of PFAS in soil and sediment. The diagram illustrates the pathway by which PFAS, originating from various sources, ultimately infiltrates to soil, sediment, and groundwater.

within the unsaturated zone to groundwater [6]. Furthermore, leaching is also regulated by the texture of the soil and the existence of heterogeneities, including root canals, the retention properties of the clay layers of PFAS and other associated factors.

The release of PFAS from contaminated soils into groundwater is influenced by geochemical factors. Longer chain and zwitterionic PFAS are often firmly absorbed in the soil; however, changes in pH for soils in AFFF-impacted areas or other sites, especially an increase in pH, could release sorbed PFAS [75,76]. Furthermore, Nguyen et al. (2020) [77] showed that the sorption coefficient (K_d values) of PFAS increased as the pH of the solution decreased from around eight to three. According to Nguyen et al. (2020), the soil-water partitioning coefficients ($\log K_d$ values) of PFAS ranged from - 0.7 to 3. However, PFAS with carbon chain lengths ranging from C_3 to C_5 had similar K_d values. This suggests that the length of their carbon chains does not influence the sorption of these substances in the soil in this case. Instead, it is likely that the binding is mainly influenced by the interaction between the ionic groups of PFAS compounds and the surface of the soil [78]. Additionally, shorter-chain PFAS exhibit a preference for the aqueous phase within the soil, making them excessively mobile. In different values of K_d in soils (0.5 to ~1.5 log units), the increases with the length of the perfluorinated C chain (C_6), suggesting that hydrophobic effects along with surface complexation with neutrally charged organic surfaces in soil may be the main sorption mechanisms for PFAS of longer chain. Furthermore, these coefficients showed a strong linear correlation with molecular weight (MW) for PFAS compounds with MW > 350 g/mol [75,77]. In general, PFAS had a higher affinity for soils with a higher organic carbon content, a higher clay and silt content, higher levels of extractable iron and aluminum, and were located in surface layers compared to subsurface horizons [77,79]. Several studies have previously highlighted the considerable influence of organic carbon (OC) on the sorption of PFAS [77,80,81]. Furthermore, Li et al. (2019) [82] suggested that the protein composition of soil organic matter could serve as a more accurate indicator of PFAS sorption compared to organic carbon (OC). However, a recent study by Campos-Pereira et al. (2023) [75] has shown that OC does not play a major role in predicting the sorption of three specific PFAS (namely, PFOA, PFDA, PFUnDA) or PFHxS. Significant information gaps persist in identifying the basic sorption mechanism of PFAS by soils, particularly when a wide range of PFAS is considered at the molecular level.

Adamson et al. (2022) [9] documented the highest recorded concentrations of PFAS in groundwater within the source areas of the three selected sites, ranging from 6 to 51 mg/L. However, they consistently observed a significant decrease in PFAS concentrations by multiple orders of magnitude as the distance from the point of origin increased at all locations. This finding suggests that natural attenuation processes effectively reduced PFAS levels along each flow path without causing significant destruction. Høisæter et al., (2019) [24] conducted an unsaturated column study using soil structure and mineralogy in a polluted location to comprehend the historical destiny of the AFFF used in firefighting areas. Consequently, PFOS was not detected in column leachate when subjected to low infiltration rates, but when introduced to high infiltration rates, PFOS was found at higher levels up to 2200 ng/L. PFOS was also observed to be highly attenuated in the unsaturated region; the infiltration rate, and the attenuation rate increases over time influence this transport. Furthermore, groundwater contamination and the spread of PFAS to neighboring surface water bodies could be potentially harmful due to the leaching of PFAS from the source area in the unsaturated region. In another study, PFOA and PFOS were meticulously evaluated in unaffected soils in the northwest direction of a fluoropolymer manufacturing facility in Dordrecht, The Netherlands, which has released PFOA since 1970 [44]. The PFOA depth patterns exhibited a discernible bell-shaped distribution, characterized by the maximum concentrations occurring at depths ranging from 20 to 50 cm below the surface. On the contrary, lower concentrations were observed at the surface and higher depths, extending up to 350 cm below the surface. The concentration of PFOS in the soil was lower, and its migration depth was lower compared to PFOA. Maximum concentrations at various depths and the consistent existence of PFOA down to a few meters suggest that PFOA exhibits mobility within these soil samples. Furthermore, the factory appears to be a significant source of atmospheric deposition of PFOA, resulting in the highest levels of this compound in the vicinity.

5. Environmental impacts of PFAS when distributed in soil

5.1. Impacts on soil microbes

The impacts of PFAS on soil microbial communities have been the subject of great interest since the activities, diversity, and composition of microorganisms are important to conserve numerous biogeochemical processes [7] and impact soil characteristics [23]. Numerous anthropogenic elements, such as PFAS, have an impact on soils around the world [3]. When living in polluted soil surroundings, organisms are susceptible to taking up organic pollutants and can accumulate excessive amounts of PFAS [31]. An experiment conducted in a greenhouse proved that PFOS (10 mg/kg) impeded soil enzyme function and subsequently damaged soil bacteria's cellular structure, an immunological system as a whole, and gene expression [83]. The different functional groups and carbon chain lengths of perfluorinated compounds (PFC) have diverse effects on soil microbe activity and can also influence the adsorption rate. In particular, sulfonic PFC showed greater toxicity compared to carboxylic ones in soil microorganisms [84]. Furthermore, an increase in chain length correlates with a higher level of toxicity in perfluorinated compounds.

The microbial activity in the soil has been associated with soil parameters such as pH, organic matter, and the availability of N, P, and K [84]. PFAS (PFOS, PFOA, and PFBS) considerably accelerated the decomposition of the litter, leading to a notable increase in soil pH, presumably as a direct result of the accelerated decomposition of the litter [3]. The accelerated decomposition may lead to increased carbon emissions, including CH_4 and dissolved organic carbon, potentially influencing soil carbon sinks [85]. PFAS also altered the amount of bacteria and fungi in the soil and greatly reduced soil respiration [3]. Qiao et al. (2018) [83] assessed the impacts of PFAS on bacterial populations and discovered that the richness, diversity, and population structure of the bacterial species were altered in the presence of PFAS. More specifically, PFAS improved the abundance of PFAS-tolerant bacteria but lowered the diversity of bacterial species [23]. It has the potential to promote the proliferation of certain bacteria while suppressing the growth of others. Therefore, exposure to PFAS interfered with soil microorganisms' ability to function normally. Furthermore, it was discovered that long-chain PFAS had more pronounced impacts on the community of bacteria in the soil than short-chain PFAS.

5.2. Impact on plants and terrestrial food web

To comprehend the transport, partitioning in environmental media, and exposure potential of PFAS in the real world, it is important to evaluate the accumulation of various PFAS derivatives in vegetation, surface, and subsurface soils. The interaction between PFAS in the soil and plant system is influenced by the types of PFAS, soil characteristics, plant physiology, and other environmental variables [17]. In the vicinity of a fluoropolymer manufacturing facility in Ohio, the average concentration of perfluoroalkyl carboxylic acids (PFCA) on the soil surface, earthworms, roots, grass leaves, and tree leaves were 150, 420, 61, 68, and 430 ng/g of (dw), respectively [86]. At concentrations >800000 ng/g, PFOA altered the germination rate, as well as the growth of the roots and shoots of wheat; however, in quantities 200 ng/g, PFOA was found to stimulate the development of wheat seedlings and root length [87]. Furthermore, the overall concentrations of PFAA in soil and wheat roots, straws, husks, and grains increased as biosolids were applied more

frequently [88]. Recently, Sivaram et al. (2023) [33] examined the absorption and accumulation of PFAS in *Allium cepa* (onion) from biosolid-amended soils. The study indicated a considerable transfer of PFAS from roots to stems, as evidenced by the positive correlation between the amount of PFAS in stems and the corresponding concentration in roots. However, the accumulation of PFAS in *A. cepa* was expected to pose a negligible risk to consumers throughout the food chain [33]. Similarly, increased levels of PFAS have been identified in Minnesota soil and locally grown vegetables due to decades of industrial waste disposal. PFAS was also found in water used for lawn and garden irrigation [39]. PFBA was detected at the highest concentrations in the produce, and the levels of PFBA in vegetables increased with the water load.

Due to the application of biosolids to agricultural land, earthworms accumulate PFOS and PFOA [31,89]. In soils impacted by biosolids, the total amount of PFAS in earthworms ranged from 9.9 to 101 ng/g (dw), and mean bioaccumulation factors (BAF) were observed between 2.2 and 98 [89]. Due to the fact that earthworms are a significant source of prey for a variety of bird species and tiny mammals, the large amounts of PFCA discovered in worms raise the possibility that these chemicals may enter the food chain. High concentrations of PFOA in the leaves of deeply rooted plants, compared to grasses with short roots, may indicate groundwater transmission from polluted aquifers. The chemical ratios between soil and grass roots and soil and earthworms revealed a positive correlation with the length of the fluorocarbon chain [86]. The presence of PFAS was confirmed by investigating sediment, water, OM (organic matter), and aquatic organisms at all levels of the food web [90]. Biofilm, which is an accumulation of bacteria, fungi, algae, and protozoans, as well as a fundamental resource for the aquatic food web, showed a significant accumulation of PFAS, in particular, for perfluorooctanoic acid, which had the highest mean concentration of 463.73 ng/g [90]. In summary, the introduction of PFAS into the environment through various pathways, such as soil, sediment, or water, leads to its subsequent accumulation in plants, vegetables, and other organisms (Table 2). Consequently, these compounds enter the food chain through trophic transfer mechanisms.

6. Environmental impacts of PFAS when distributed in sediment

6.1. Accumulation and fluxes of PFAS in sediment

The study of sediment cores has proven to be a valuable method for assessing historical pollution associated with exposure to PFAS. The accumulation of PFAS in sediments is primarily influenced by the physicochemical characteristics of both the PFAS compound and the sediment and the quality of the sediment. In general, long-chain sulfonic acids tend to exhibit the highest levels of sorption among PFAS compounds [62,91]. Therefore, with increasing chain length, the distribution coefficients (K_d) of PFAS within sediment and water increased [92], and $\log K_d$ and $\log K_{ow}$ (octanol-water partition coefficients) exhibited a positive correlation [91]. For example, in the Truckee River and the Las Vegas Wash in Nevada, USA, the total PFAS concentration in the sediments was 272.9 ng/g (dw) and 345.7 ng/g, respectively. The results also showed that short-chain PFAS ($C \leq 8$) were more common in the water, whereas long-chain PFAS ($C > 8$) were more noticeable in the sediments [70]. The $\log K_{oc}$ values for C_8–C_{12} PFCA in sediments from Dianchi Lake in China varied between 2.54 and 3.57 [93]. Moreover, the movement of PFAS in waterways can be affected by its partitioning to sediment. However, the degree of partitioning is determined by various factors, such as the length of the PFAS chain, the chemistry of its head group and the disparities in water and sediment chemistry [94,95]. Organic matter in sediment is likely attracted to long-chain PFAS molecules due to their increased hydrophobicity and sorption affinity [70].

Investigation of sediment cores collected from lakes and ponds in Sweden in 2015 revealed that PFAS contamination began in 1994 near a firefighting facility. Maximum accumulation of PFAS was recorded during the period 2003–2009. PFAS flux was found to increase from 2.3 μg/m^2/year (dw) in 1994 to 12 μg/m^2/year (dw) in 2009 [96]. Additionally, a decrease in the magnitude of sediment accumulation was observed with increasing depth. Specifically, accumulation rates were measured at 9.2 kg/m^2/year (2–4 cm) and 8.4 kg/m^2/year (4–6 cm). These rates continued to decline to a range of 0.29–0.31 kg/m^2/year (6–12 cm). Another study found an increase in PFOA, PFDA, PFBS, and PFOS fluxes from the sediment core in Lake Hazen, Canada, between 1963 and 2011. Similarly, the PFHpA, PFOA, PFNA and PFUnDA fluxes increased in Lake B35 between 1952 and 2009 [53]. The patterns seen in the sediment cores indicate temporal changes that parallel variations in the historical level of PFAS production, suggesting that there may be a link between the two. The abundance and temporal pattern of PFAS in the water and sediment layers correlate with historical changes in the inventory of PFAS manufacture. Furthermore, the possibility of contamination discharge due to factors induced by climate change is a potential concern. Historical records of PFAS monitoring do, nevertheless, present certain challenges. For example, ensuring storage integrity during long-term environmental monitoring can present logistical challenges and involve significant costs. Additionally, the evolution of analytical methodologies over time can complicate the quantification of temporal trends and impact interannual assessments [53].

6.2. Impact on sediment microbes and benthic microorganisms

Sediment in water reservoirs plays a crucial ecological role, provides

Table 2
Environmental impact of PFAS on biotic and abiotic components (ng/g, dw).

PFAS types	Interactive Media	Concentration	Impact	References
[b]ΣPFCAs	Soil-earthworm	420	PFCA accumulation in earthworms	[86]
ΣPFCAs (C_8–C_{14})	Soil-earthworm	9.88–99.86	Total accumulation of PFAS in earthworms	[89]
[b]ΣPFCAs	Soil-plant	61	PFCA accumulation in grass root	[86]
[b]ΣPFCAs	Soil-plant	68	PFCA accumulation in grass leaf	[86]
[b]ΣPFCAs	Soil-plant	430	PFCA accumulation in tree leaf	[86]
[a]ΣPFAAs	Soil-plant	140–472	PFAA accumulation in roots	[88]
[a]ΣPFAAs	Soil-plant	36.2–178	PFAA accumulation in straws	[88]
[a]ΣPFAAs	Soil-plant	6.15–37.8	PFAA accumulation in husks	[88]
[a]ΣPFAAs	Soil-plant	7.32–35.6	PFAA accumulation in grains	[88]
PFOA, PFBA	Soil	100	Significantly increased litter decomposition and soil pH	[3]
PFOA, PFBS	Soil	500	Decreased soil respiration	[3]
PFOA	Soil-plant	>800000	PFOA restricts the germination rate and root and shoot growth of wheat	[87]
PFOA	Soil-plant	>200000	The height of the wheat seedling plant height was inhibited	[87]
ΣPFAS	Soil-Plant	10.4–104	PFAS accumulated and transferred from the soil to *the* roots and shoots.	[33]
PFOS	Soil-bacteria	1000	The richness, diversity, and population structure of the community of bacterial species altered.	[83]

[a] ΣPFAA = PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFUnA, PFTeA, PFBS, PFHxS, PFOS.
[b] ΣPFCA = PFHpA, PFOA, PFNA, PFDA, PFUnDA, PFDoDA

a habitat for a wide variety of benthic species [21], and serves both as a temporary or long-term contamination sink [69,97] for substances such as PFAS, which pose a threat to benthic species [98]. Therefore, sediment plays an important role in aquatic habitats, significantly influencing the well-being of organisms and communities (Table 3) [19]. The widespread presence of PFAS in sediment is a growing concern, often associated with specific origins, such as firefighting foam or industrial operations. The discharge of PFAS from sediments into the above water column could have detrimental toxic impacts, affecting both aquatic creatures and human health in cases where a river or lake serves as a drinking water source [99,100].

Sediment microbe populations drive many chemical biotransformation activities in aquatic habitats and play an important role in various ecosystems and biogeochemical cycles [21,104]. The substantial presence of PFOA in the sediments likely altered the composition of the microbes, introducing an increased abundance of *Proteobacteria, Thiobacillus*, and *Sulfurimonas* [28]. Yun et al. (2023) [101] investigated how organic carbon concentrations affected PFAS bioaccumulation in benthic creatures such as worms, mussels, and snails. The findings revealed a considerable variation in PFAS bioaccumulation, with worms exhibiting significantly higher levels compared to snails and mussels, indicating a species-dependent variation. According to Wang et al. (2023) [5] 13 PFAS were found in a sediment-water system with known amounts of PFAS. These chemicals were found to be harmful to *Daphnia magna*, zebrafish, and the benthic animal *Chironomus plumosus.* Rising temperatures made it easier for PFAS to move between sediment and adjacent water levels, increasing the quantity of PFAS in the water and decreasing its concentration in the sediment. As a result, pelagic creatures such as zebra fish and *D. magna* had a higher body burden. Pulster et al. (2022) [102] found that PFAS levels in sediment and consumable fish tissues near Tampa Bay, USA, ranged from 0.036 to 2.99 ng/g (dw) and 0.3–33.6 ng/g (ww), respectively. Therefore, concerns were raised about the high concentrations of PFAS and the fact that they exceed the established limits for potential dangers to human health.

Zhang et al. (2017) [104] examined the influence of varying concentrations of 6:2 fluorotelomer alcohol on the composition of a sediment-based microbial population. They discovered that the addition of 6:2 FTOH and the formation of intermittent transformation substances in the sedimentary layer created selective pressure on the range of species and taxonomic distribution of microorganisms. Their findings also suggested that bacteria could decompose 6: 2 FTOH, including *Dokdonella* spp. and *Thauera* spp. *Albidovulum* spp. and *Caldanaerovirga* spp. were present in the sediment mixture and gradually became predominant. Furthermore, the high level of PFOA (456.20 ng/g (dw)) has the potential to limit the diversification of the bacterial community. When the total concentration of PFOA was less than 100 ng/g, there was no discernible influence on the structure of the microbial community [28]. However, little effect on the composition of the microbial population was observed when the total concentration of PFOA in the sediment was less than 100 ng/g (dw).

Table 3
Environmental impacts of PFAS on benthic organisms through sediment exposure (ng/g dw).

PFAS types	Interactive Media	Concentration in biota	Impacts	References
PFOA	Sediment-bacteria	–	Decreased bacterial community diversity	[28]
Σ_{14}PFAS	Sediment-Benthic Organism	2216.7	Bioaccumulation in worms	[101]
Σ_{14}PFAS	Sediment-Benthic Organism	61.7	Bioaccumulation in mussels	[101]
Σ_{14}PFAS	Sediment-Benthic Organism	21.1	Bioaccumulation in snails	[101]
Σ_{25}PFAS	Sediment-fish	0.036–2.99	Fish contamination and human health risk	[102]
Σ_{7}PFAS	Sediment – Benthic organism	31.65	Bioaccumulation of PFAS in mayflies	[103]
Σ_{7}PFAS	Sediment – Benthic organism	27	Bioaccumulation of PFAS in Hellgrammites	[103]
Σ_{7}PFAS	Sediment – Benthic organism	13.4	Bioaccumulation of PFAS in Water Pennies	[103]
Σ_{7}PFAS	Sediment – Benthic organism	11.37	Bioaccumulation of PFAS in Caddisflies	[103]

7. Measures to limit the distribution of PFAS in soil and sediment

In general, PFAS are not degraded by microbes, leading to their persistence in the soil [105]. Furthermore, PFAS accumulated in soils can impact ecosystem functions [3]. A significant portion of the PFAS pollution in the receiving waterways comes from runoff in tributaries. Furthermore, the composition of PFAS in these tributaries has been linked to potential sources, including firefighting foam, various industrial operations, and airports [38]. Effective management of solid waste through treatment and incineration is vital to safeguarding the environment and mitigating the potential hazards associated with toxic exposures to PFAS. Therefore, effectively addressing the issue of PFAS requires control over many potential sources. These sources include wastewater runoff, atmospheric emissions, biosolid application, appropriate recycling of consumer products, proper disposal of PFAS waste, and ultimately, prohibition of PFAS manufacturing through the enactment of laws. To ensure the efficacy of these measures, rigorous monitoring and management practices must be implemented. For example, following the Stockholm Convention resolutions, the Pearl River region of China implemented efficient management and regulatory measures to address the pollution generated by the PFAS sectors, resulting in a decrease in the concentration of PFAS in sediment [21]. Presently, the United States does not have nationally enforceable regulations for PFAS in soil, which requires individual states to manage a variety of strategies to address PFAS contamination [106]. However, on March 14, 2023, the US EPA published a proposal for a National Primary Drinking Water Regulation (NPDWR). This regulation aims to set legally mandated maximum contamination levels (MCLs) for six PFAS present in drinking water only [107]. On the other hand, Australia, Denmark, Germany, the Netherlands, and Sweden have all established PFAS limits for soils. No other nation was found to have imposed PFAS limits on biosolids or their use [108]. However, only the state of Maine in the United States has a law prohibiting land use of biosolids, and a few states have passed or suggested laws to control PFAS in these materials [109]. Recent emission inventory data associated with PFAS from fluoropolymer production facilities suggest that the discharge of these substances into the environment does not exhibit a declining trend and instead maintains a relatively stable level in Europe [110]. The Environmental Protection Agency (EPA) published a PFAS action plan in February 2019, but it lacks fundamental information on these compounds' toxicities, exposure pathways, and remediation approaches. It also does not specify how the EPA will change its toxicity assessment strategy to ensure that new PFAS remain out of the market [111].

Furthermore, it could be anticipated that the use of the in-situ stabilization and solidification (S/S) technique will serve as a financially viable and environmentally sustainable approach to managing source zones already affected by AFFF. This method involves the application of a chemical amendment to mitigate or eliminate contamination leaching [112]. The application of stabilizing amendments consistently showed reductions in the leachability of PFAS by more than 99 % on a bulk basis.

However, there is the possibility that PFAS is reintroduced into the environment as the stabilizing materials undergo degradation [113]. Implementing a prospective nationwide prohibition on the application of biosolids on agricultural land would reduce PFAS in groundwater. This, in turn, has implications for drinking water quality, since groundwater serves as a potential source of drinking water. Management of atmospheric deposition poses significant challenges and may be effectively addressed solely through the implementation of source control measures. Efforts must be made to research and determine the presence of PFAS dispersed previously in soil or sediment close to areas impacted by these substances. Subsequently, appropriate remediation methods must be implemented to prevent the continued dissemination of PFAS through surface and groundwater.

8. Challenges and future perspectives

Numerous studies have shown that PFAS from soil has entered the terrestrial food web. Identification, assessment, and understanding of PFAS absorption in soil environments will provide crucial details on exposure routes and assist in designing adaptable responses. Davis et al. (2023) [8] recently found that plants and soils in New Jersey accumulated chloroperfluoropolyethercarboxylates (Cl-PFPECA) and perfluorocarboxylates (PFCA). They also found that these accumulation factors (AF) increased as the CF_2 count decreased. Because the production of PFAS has switched from long-chain to short-chain chemistry, more short-chain PFAS is building up in plants. This means that people and wildlife habitats worldwide may be exposed to levels that were not expected. Furthermore, this replacement may influence drinking water quality through groundwater contamination, as shorter chain variants tend to exhibit greater mobility into deeper soil layers. This increased penetration increases the likelihood of groundwater contamination [114]. The presence of PFAS in soils has been observed in areas that are geographically distant from recorded or recognized sources of PFAS contamination. In particular, sample locations are often located several hundred kilometers from urban or industrial areas that are known to be associated with PFAS production. This finding provides evidence for long-range atmospheric transport of PFAS compounds [6]. Therefore, the dispersion of PFAS in the air from industrial operations and subsequent global atmospheric transport pose a major hurdle in the management of soil contamination. PFAS was found in large amounts in most types of benthic macroinvertebrates and in the soil. However, PFAS levels in sediment and surface water were insignificant in the same area. This means that monitoring only abiotic matrices may not give an accurate picture of how polluted local aquatic environments are [103].

The commercially viable options for addressing source zones affected by AFFF are currently limited to in situ stabilization, excavation with off-site disposal or destruction off-site, and on-site treatment by soil washing or thermal desorption. However, in areas where AFFF is present, implementing measures such as adding an adsorbent or complexing agent to halt the movement of PFAS in the saturated zone may pose challenges. This is especially true regarding how chemicals are spread in subsurface environments [112]. Current soil remediation techniques, including thermal treatment to remove PFAS from soil, chemical oxidation, membrane filtration, and bioremediation, face significant hurdles in terms of scaling up operations in the field, substantial cost, limited efficacy in removing PFAS, and incomplete elimination of these contaminants [113]. The development of innovative treatment technologies that can effectively remove PFAS on an industrial scale is both critical and challenging. To effectively mitigate the extensive contamination of PFAS compounds within a given region, conducting a thorough investigation that includes many environmental components is imperative. This comprehensive study must include the analysis of soil, sediment, surface water, groundwater, benthic invertebrates, fish, and plant species.

9. Conclusion

This research discusses the sources that contribute to the contamination of soil, sediment, and groundwater, given the interconnectedness of these three media. Soil and sediment function as long-lasting repositories for persistent organic pollutants, such as PFAS [115]. The extensive presence of PFAS in soil has been thoroughly documented, suggesting that the ecological consequences are also long-lasting in character. The results of this review related to the origin, dispersion, and consequences of PFAS in soil and sediment are listed below.

1. The primary sources responsible for the contamination of PFAS in soil and sediment include the use of aqueous film-forming foam during firefighting operations, air emission and discharge of PFAS from fluoropolymer production facilities, application of biosolids in agricultural practices, leaching and emission of PFAS from landfills, industrial discharges, incineration of waste in landfills, irrigation water, and wet and dry deposition of PFAS from the atmosphere. Based on the sources analyzed, it is evident that aqueous film-forming foam, fluoropolymer production facilities and biosolids are the primary sources responsible for substantial pollution.
2. The possible risk of groundwater pollution and subsequent impacts on organisms dependent on this water source may arise from the downward movement of short-chain PFAS from the uppermost layer of the soil. In light of this observed occurrence, coupled with the increasing use and production of short-chain substitutes, it is crucial to conduct further research concerning the adsorption, transportation, and ecological consequences of PFAS in deeper soil strata.
3. Currently, there is a lack of data on the effects of PFAS on soil composition and microbially driven activities. However, the presence of PFAS in soil has been reported to adversely affect cellular structure, immune system, and gene expression and alter soil bacteria's richness, diversity, and population structure. Additionally, PFAS exhibited a notable acceleration in the litter decomposition process, leading to a considerable increase in soil pH. Furthermore, PFAS can accumulate in the soil and subsequently translocate to both the roots and the shoots of plants.
4. Microbes and benthic organisms can be affected when PFAS are present in sediment. A more comprehensive understanding of the effects of PFAS in sediments could be achieved by expanding the types of organisms studied in future environmental impact assessments.
5. To effectively address the issue of PFAS, it is imperative to exercise control over many potential sources, as discussed. Efforts must be made to research and determine the presence of PFAS previously dispersed in the soil or sediment. Consequently, effective remediation strategies must be implemented to stop the further spread of PFAS through surface and groundwater. It is crucial to perform a comprehensive assessment that includes several environmental factors, such as soil, sediment, surface water, groundwater, benthic invertebrates, as well as fish and plant species.

CRediT authorship contribution statement

Mohammad Nazmul Ehsan: Writing – review & editing, Writing – original draft, Visualization, Formal analysis, Data curation, Conceptualization. **Mumtahina Riza:** Writing – review & editing, Writing – original draft, Formal analysis, Data curation, Conceptualization. **Md Nahid Pervez:** Writing – review & editing, Writing – original draft, Formal analysis, Data curation. **Chi-Wang Li:** Writing – review & editing, Writing – original draft, Supervision. **Antonis A. Zorpas:** Writing – review & editing, Supervision. **Vincenzo Naddeo:** Writing – review & editing, Supervision, Project administration, Funding acquisition.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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